Evaluating Sources of Trace Element Contamination in New York State Urban Vegetable Gardens by Analysing Paired Soil-Vegetable Samples

Salvatore Engel-Di Mauro

Abstract
Trace element (TE) contamination in urban vegetable gardens (UVGs) involves many sources often difficult to distinguish one from the other. Data from a 2013-2014 study were re-analysed to identify soil-borne sources of vegetable TE contamination. The study included paired surface soil (0-15 cm) and vegetable sampling (38 pairs). Preserved samples (n=22) were re-analysed in 2016-2017 for extractable Mg and Pb. Average soil Pb and Zn and vegetable TE levels were above safe levels. Vegetable TEs were unrelated to soil properties, except Ni and %SOM. Hg, absent in soils, contaminated vegetables in UVGs within the same neighbourhood. Soil-vegetable TE correlation was significant only for V. Vegetable Al positively correlated with vegetable Cd, Cu, Hg, Pb, Ti, V, and Zn. Extractable Pb was low compared to Mg, with no relation to other variables except between extractable Mg and %SOM. Grouping data according to planting method showed raised beds did not prove effective in preventing contamination. Results for Hg and positive vegetable Al-TE correlations suggest contamination mainly from sources other than soils. The findings indicate that paired soil-vegetable sampling helps differentiate soil-borne TE from other sources. This helps formulate more UVG-specific analyses and exposure reduction measures.

Keywords: heavy metals, soil contamination, trace elements, urban soils, urban vegetable gardens

Introduction
Urban vegetable gardens (UVGs) have had a major resurgence (Mok et al., 2014; WinklerPrins, 2017), but there is concern over trace element (TE) contamination (Attanayake et al., 2014; Cheng et al., 2015; Hough et al., 2004; Hendershot and Turmel, 2007; Spittler and Feder, 1979; Wortman and Lovell 2013) as well as ambiguities regarding contaminant sources (Brown et al., 2016; Clark et al., 2008). Soil particle inhalation or ingestion and consumption of contaminated vegetables constitute main health-menacing pathways (Brevik and Burgess, 2016; Mitchell et al., 2014). TE contamination sources in UVGs include former land use legacies, anthropogenic and geogenic characteristics of underlying strata, and long-range and nearby pollution, including from vehicular traffic (Alloway, 2004; Jean-Soro et al., 2015; Pouyat et al., 2007; Säumel et al., 2012; Sung and Park, 2018; Zwolek et al., 2019).

Many studies focus on TE transfer from soils to vegetables, showing how soil-borne vegetable contamination is governed by multiple inter-related factors that include human activities (Alloway, 2013; Cheng et al., 2015; Gupta 2019; McBride, 1989; Menefee and Hettiarachchi, 2018; Tack, 2010). Gardeners' inputs can introduce TEs directly or alter soil properties in ways that favour or mitigate vegetable TE absorption (Agbenin et al. 2010; Bolan and Duraisamy 2003; De Miguel et al. 1998; Nabulo et al. 2010). Land-use legacies can also attenuate TE plant availability (Howard and Olszewska 2011). Atmospheric deposition and local re-deposition complicates soil-plant interactions and TE contamination processes (De Temmerman et al., 2015; Fismes et al., 2005; Rajput et al., 2017; Xiong et al., 2014). Consequently, TE levels in vegetables do not necessarily correspond with those in the soil on which they grow (Allen and Janssen 2006; McBride et al. 2014; Warming et al. 2015).

More research is needed to discern the relative importance of contamination sources, especially atmospheric deposition. Each UVG also presents unique

characteristics that known processes and mechanisms may not capture adequately for hazard prevention purposes (Brown et al., 2016; Hursthouse and Leitão, 2016; Menefee and Hettiarachchi, 2018). Adding to this is the possibility of significant changes resulting from evolving and mutually transforming interactions between soil properties (e.g., pH, redox conditions), cultivation practices (including gardener inputs), and external contaminant sources (e.g., changes in vehicular traffic burden). Raised bed soil Pb recontamination (Clark et al., 2008) is but one indication that UVG contamination varies both spatially and temporally (McBride, 1995).

Yet in many studies, contamination is assumed to be traceable mainly to root absorption (e.g. Attanayake et al., 2014; Brown et al., 2016; Dala-Paula et al., 2018; Friol Boim et al., 2016; Gu et al., 2018; Gupta et al., 2019; Li et al., 2018; Rehman et al., 2018). The assumption often appears in the guise of bioconcentration factors, computed by dividing plant TE content by the counterpart in soil. What is more, studies sometimes exclude either vegetables or soils, or soil and vegetable samples are spatially disconnected. Not accounting for or differentiating among environmental variables may lead to over-estimations of soil-borne TE contributions, if not a misidentification of the main sources of contaminants other than soil. The problem is only exacerbated by spatially disconnected soil-vegetable sampling.

One way of reducing the analytical complexity is by differentiating soil-borne from other vectors. It can serve as a preliminary step towards detecting TE contamination sources, including for UVG-specific purposes. This can be important when accounting for such a multitude of factors can be too burdensome for most communities (this is without even considering the social power relations affecting UVG contamination variables; see McClintock, 2015). To contribute towards primary source differentiation, paired soil-specimen data from a previous project on urban community gardens in New York State (Engel-Di Mauro, 2019) were re-analysed statistically, including by examining vegetable Al-TEs correlation (McBride et al., 2014; Paltseva et al., 2018). Preserved soil samples were also processed for extractable Mg and Pb. The objective was to assess the extent of non-soil sources of vegetable TE contamination useful towards source differentiation (i.e., soils vs. external fluxes) and based on relatively routine laboratory testing.

Materials and Methods

UVGs were contacted and sampled in four cities of New York State (USA), Albany (two), New York City (three in Brooklyn and eight in Manhattan), Syracuse (five), and Troy (four), for a total of 23 UVGs (Table 1). A parcel in a peri-urban farm in Delmar, near Albany, was also sampled for comparison and designated as garden F. Planting in that farm section was on existing soil. Albany UVGs include one community garden and one experimental and educational garden. All New York City and Syracuse UVGs are community gardens, one of which (in Manhattan) is exclusive to residents of a high-rise building. Save for a shared backyard garden, Troy UVGs are community gardens.

No agrochemicals were used in UVGs and in the peri-urban farm, but different kinds of compost were applied at all sites. Planting method was split between use of pre-existing soil and of raised beds comprised of imported “soil” (Table 1). All Albany and Troy gardens were planted directly on original soil, in contrast to those in New York City, which consisted of raised beds. The planting techniques in the Syracuse gardens were split between two using raised beds and three using existing soil.

Within each UVG, one to four harvestable plants were removed whole along with soil (0-15 cm depth) within a 5 cm radius. This allowed for direct pairing of plant and soil samples (38 pairs). Sample numbers varied according to UVG heterogeneity.

More than one sample was taken from sites of larger area and exhibiting greater variability, namely in terms of duration of use for various areas, differences in land use history prior to urban garden conversion, and physical layout. The same sampling rationale (two crop-soil pairs) was applied to a fenced area of comparable size to UVGs in the Delmar farm.

All samples were collected between 26 July and 7 August 2013. Soil sampling equipment was washed with deionised water and wiped clean with paper towels after each use. At least 2 kg of soil was placed in a sterile plastic bag for each sample. Specimens were selected according to gardeners' priority, known relative bioaccumulation potential, and harvestability during the fieldwork season. The plants included a variety of herbs, leafy greens, and fruiting vegetables, with one root vegetable (radish) and one cucurbit (Table 1). Planting method was on raised beds and directly on soil for 24 and 14 crops respectively. Each specimen was dusted prior to placement in the sterile paper bags to exclude airborne-derived contaminants and prevent fungal infestation. Vegetables were washed with deionised water and air-dried in a drying oven at the Soils Lab in SUNY New Paltz.

Within 48 h of collection, vegetable specimen and soil samples were posted to the Cornell University Nutrient Analysis Lab. Soil samples were analysed for particle size distribution, % soil organic matter (% SOM, LOI method), pH (water). Particle size analysis failed to be reported for one of the soil samples at garden G. Pseudo-total TE content was analysed for both soils (ICAP Elements Hot plate $HNO_3/HClO_4$ digestion) and plant tissues (hot plate digestion and ICP/AES HM analyses). Samples were tested for a suite of elements, including, for the purposes of this re-analysis, Mg and the trace elements Al, As, Cd, Co, Cr, Cu, Hg, Ni, Pb, Ti, V, and Zn. Tests for As were found reliable only for soil samples. Cr was not differentiated according to hexavalent and trivalent species. Preserved soil samples (n=21) were re-analysed in 2016-2017 at the Analytical Chemistry Lab of SUNY New Paltz for extractable Mg and Pb (FAAS; $CaCl_2$ method). Mg was used for comparative purposes between macro-nutrient and TE fluxes.

Due to the different aims of the original investigation, no fresh weight was recorded for the plant specimens. Conversion factors (0.2 for root vegetables; 0.25 for leafy vegetables; and 0.18 for fruit) were therefore used (Staven et al., 2003) to allow comparisons with established food quality standards (FAO/WHO Codex Alimentarius Commission, 2015). This kind of estimation tends to err on the permissible side, but this can be justified, aside from the limited objectives of this study, by the restrictive assumptions of 100% intake bioavailability made in the Codex Alimentarius (Defoe et al., 2014). Accordingly, the lower (more stringent) vegetable limits of the Codex Alimentarius food quality standards were used for comparisons with plant specimens' TE content relative to their calculated fresh weight values. Minimum Risk Levels provided by the US Center for Disease Control (US CDC, 2019) were used for TE thresholds not provided in the Codex.

Data were subjected to descriptive statistical measurements, correlation analyses (Pearson's r), arithmetical transformation, linear regression analyses, one-way ANOVA, and independent-samples t-tests, all using Microsoft 10 Excel and IBM SPSS Statistics 20. New York State Department of Environmental Conservation soil screening standards were used to interpret results on TEs where applicable. Otherwise, US EPA information was used. Shapiro-Wilk tests were used to identify data sets departing from Gaussian characteristics. Relationships between variables were inspected through scatter plots and correlation analyses (D'Agostino and Stephens, 1986), focusing on soil and plant specimen TE content and especially on crop Al relative to crop TE content.

Cases of divergence between soil and crop TE data were highlighted by subtracting the latter from the former. Where tenable, linear regression analysis was applied to soil and plant TE data to test for potential patterns relatable to soil-derived contamination. Linear regression was also extended to analyse crop Al content as predictor variable on other crop TE content. This was to inspect the possibility of contamination by means of soil particle re-deposition (e.g., splash) and atmospheric deposition with foliar uptake and particle lodging (Xiong et al., 2014). Additionally, data were grouped according to cultivation technique (on pre-existing soil or on raised beds). For grouped data unsuitable for one-way ANOVA (e.g., unequal variance, inapplicability of post-hoc tests), independent-samples t-tests were applied (Timm, 2002). These statistical tests were carried out to examine the possibility of there being effectively two statistical populations and to detect effects of cultivation method on soil TE concentrations. The expectation was that raised beds would be significantly less contaminated and, if so, any plant contamination would likely be traceable to sources beyond soil.

Results

Soil characteristics tended towards slightly acidic on average, with highly variable SOM and clay content (Table 2). According to available New York State soil quality standards for unrestricted land use (NYS DEC 2006), levels were below safety concerns only for As (Table 3). Average values were within safe levels for Cd, Cu, Hg, Ni, but not Pb and Zn. For the remaining TEs, standards have not been established in New York State, but relative to US EPA screening procedures (US EPA 2017), Co and V would be within safe levels. Results for Cr could not be evaluated due to the lack of lab-analytical species differentiation. No maximum allowable levels could be found for Ti.

Plant specimens exhibited high average amounts of TEs beyond safe oral intake levels for Al, Co, Cr, Hg, Pb, V, and Zn (Table 4). Maximum values exceeded safe levels for all TEs for which standards have been established. Eight vegetables on raised beds were found to contain levels of Hg between 0.62 and 3.52 mg/Kg. The Hg hazard applies only to Lower East Side Manhattan gardens and it was the only instance where TE contamination had a clear geographical delimitation. In fact, except for Sr, the highest levels of vegetable and herb contamination (sites H, J, L) were found to occur in that part of New York City (Table 5).

However, no significant correlations were found between plant TE content and the soil properties examined (pH, % SOM, and % clay). Soil Hg was absent in soils but detected in six vegetables (1.363-3.520 ppm). No significant correlations were found between soil and crop TEs, except for V (Pearson's $r = 0.758$, $p < 0.001$, $df = 37$). For soil and crop Mg, there was an inverse correlation (Pearson's $r = -0.400$, $p < 0.05$, $df = 37$). Significant positive correlations were instead found in plant samples between Al and seven of the 11 TEs considered, that is, Cd, Cu, Hg, Pb, Ti, V, and Zn (Table 6).

Extractions for exchangeable cations revealed greater mobility for Mg compared to Pb. Unlike exchangeable Mg (5.383-6.859 mgL^{-1}), exchangeable Pb was low (0.098-0.356 mgL^{-1}). Save for a positive correlation between % SOM and extractable Mg, no relationships were found between extractable Mg and Pb and soil properties or plant TE content.

Crop TE levels exceeded soil TE concentrations in 39 instances for the elements considered in this study, or 8.6% of the total (Table 7). The figure for Hg is noteworthy because no Hg was detected in soil. Consistent with the above-described findings, all cases of Hg contamination were in UVGs of the Lower East Side of Manhattan. Correlation analyses revealed that Hg was significantly associated with crop Al (Pearson's $r = 0.411$, $p < 0.05$, $df = 37$), Cu (Pearson's $r = 0.726$, $p < 0.001$, $df = 37$), Pb

(Pearson's $r = 0.649$, $p < 0.001$, $df = 37$), Zn (Pearson's $r = 0.420$, $p < 0.01$, $df = 37$), but negatively correlated with Cr (Pearson's $r = -0.391$, $p < 0.05$, $df = 37$).

When data are grouped according to planting method some of the above-described patterns come into greater prominence and new ones emerge between soil properties and element concentrations (Table 8a-b; statistical figures for As and Hg are omitted because they were not detected or reported for one or another variable). Some of the differences were found to be significant. The results of Levene's tests showed that data characteristics for soil Cr, Cu, Ni, Sr, Ti, V, and Zn and crop Cd, Cr, Pb, and Sr conformed to assumptions for one-way ANOVA. Of these groups, the use of raised beds made a difference only for soil Ti ($F = 13.896$, $p < 0.001$, $df = 37$). The independent-samples t-tests on the rest of the planting-technique groupings showed significant differences for soil Al ($F(17.749) = 9.904$, $p < 0.05$) and Co ($F(16.521) = 8.059$, $p < 0.001$). There were also significant differences with crop Cu ($F(23.560) = 7.129$, $p < 0.05$), Hg ($F(23.000) = 24.271$; $p < 0.05$), and Zn ($F(28.516) = 6.148$, $p < 0.01$).

Given the above-described results of correlation analyses, linear regression analyses on soil and crop data yielded no significant outcome except in the case of soil and crop V and crop Al and seven TEs (data not conforming to Gaussian distributions were normalised using Blom transformation). Accordingly, it was found that soil V could explain 56.2% of crop V variance ($F(1, 36) = 48.561$; $p < 0.001$). Crop Al content could only explain 37.2 % of crop Cd ($F(1, 36) = 22.923$; $p < 0.001$) and 11 % of crop Hg ($F(1, 36) = 5.564$, $p < 0.05$). But it showed greater influence over crop Pb (60.5 %; $F(1, 36) = 57.761$; $p < 0.001$), Ti (75.1 %; $F(1, 36) = 112.591$, $p < 0.001$), and V (70.3 %; $F(1, 36) = 88.458$, $p < 0.001$).

Discussion

Re-analysing paired soil-vegetable samples helps as a preliminary step towards disentangling the multiple sources of vegetable TE contamination, on the basis of known mechanisms. Major soil characteristics included pH and % SOM levels that tend to suppress mobility for most TEs (Alloway, 2013; Allen and Janssen, 2006; Violante et al., 2010). Percentage clay, though broadly useful in setting up initial expectations, is much too generic for this kind of analysis, without, for example, any breakdown along mineralogical and consequent cation exchange characteristics. While soil TE contamination was substantial and concentrations often above recommended limits, the overall impression even from such rudimentary data would already be that contamination via root absorption should be unlikely for the UVGs studied, barring soil TE saturation (McBride, 2013). This does not rule out a need for further investigation into other soil properties (e.g., iron oxides content) and processes (e.g., redox). A lack of a significant relationship between soil and plant TE levels, as found in this study, does not address the influence of other major soil variable on soil TE mobility. Low soil Pb exchangeability, for instance, could be a temporary phenomenon coinciding with the field sampling period.

Yet getting a better handle on soil dynamics would still be insufficient in the cases examined here. The occurrence of TEs like Hg in vegetables grown on soils with no detectable Hg suggests contamination pathways beyond soil. Soil Mg exhibited greater mobility than Pb. Unlike Pb, Mg appears to reach vegetables through root absorption. This can be inferred from the significant correlation discerned between vegetable Pb and vegetable Al and Hg, the latter being clearly from airborne sources. The inverse relationship between soil and plant Mg cannot be explained according to the data gathered, but there were strong positive correlations between both plant and

soil Mg and Ca and clay content. This is not reported here, as it takes the discussion beyond the scope of this study, but the matter could be explainable by the interplay of rhizosphere cation exchange mechanisms, including clay exchange site sorption processes and root absorption.

Potential macro-nutrient behaviour aside, the lack of correlation between almost all soil and vegetable TEs could suggest that much of the TE contamination is not coming from soils. The exception is V, as indicated from the linear regression analysis of soil and vegetable V content. Because of the limited nature of the soils data, it would be premature to conclude anything about the principal provenance of contaminants.

There are still other findings hinting that contaminated soil is the least contributor to vegetable contamination. Planting on original soil or raised beds with imported "soil" seems to be significant for soil Al, Co, and Ti and for vegetable Cu, Hg, and Zn content. For soils, there was an approximately tenfold difference for those TEs, possibly due to soil and imported sediment heterogeneity combined with effects from compost additions of variable TE content. More importantly, the variation in the specificity of grouped TE differences between soils and vegetables indicates that the plant specimens' TE content is inexplicable relative to soil contamination levels. For Hg, the outcome of the statistical test further underlines a likely atmospheric deposition effect on raised beds. As discussed above, there were no differences in soil Hg levels, as no Hg was detected in any soil samples. The cases of vegetable Cu and Zn contamination is also unlikely soil-borne because the means for soil Cu and Zn were similar. Hence, the result can instead be interpreted as due to other sources, such as gardener input, atmospheric deposition, and re-deposition.

Vegetable Al-TE correlations reveal additional trends. Al is generally not taken up by plants, especially at pH ranges typical of UVGs. Finding positive, significant vegetable Al-TE correlations therefore suggest contamination pathways other than soils. Soil Ti was higher in the original compared to raised-bed soils, while the opposite was found regarding vegetable Ti (Table 8b). With vegetable Al explaining 75.1% of vegetable Ti variance, it would seem that the higher vegetable Ti associated with raised bed planting technique is likely due to local re-depositional processes internal to those UVGs.

In sum, only vegetable V appear to originate from multiple sources that include contaminated soil, while the remaining vegetable TE levels are more likely from other sources. These could include gardener inputs (e.g., compost, irrigation water), hydrological dynamics (e.g., runoff from polluted sites, hurricane-induced settling of TE-laden sediment), and airborne fluxes, like long-range atmospheric influx, local intra-urban fluxes (settling of entrained particles from nearby traffic), and intra-UVG re-deposition (e.g., irrigation-induced splash). Given the overall results, high vegetable TE content should compel a study of contamination processes beyond contaminated soil. This is most evident in the case of Hg, which was undetected in soils, but in general findings point to the need for additional data gathering on sources aside from contaminated soils. Bioconcentration figures would in this case yield incorrect estimates because sources other than soil are involved in vegetable contamination.

Initial TE contaminant source differentiation can be derived through paired soil-vegetable sampling and relatively routine lab analyses, subsequently examining statistical relationships between soil properties, soil and vegetable TEs, vegetable Al and TEs, and extractable soil TEs and macro-nutrients. Because they encompass multiple UVGs, the results of this re-analysis represent a diversity of environmental contexts and outcomes of land use impacts, including different cultivation techniques that may be irrelevant to analysis at UVG scale. Nevertheless, a similar sampling design

and multiple-step statistical procedure, as shown here, can be applied to single UVGs (ideally including uncultivated surfaces as well). A methodological framework such as the one herein developed may not specify the desired contaminant source specificity, but it helps, as a first approximation, determine whether vegetable TE contamination is due to TE-contaminated soil.

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Table 1. Urban vegetable garden sampling locations and general characteristics (sampling site numbers reflect only the 38 paired soil-vegetable samples, in sample-taking chronological sequence). Identifying information is omitted to preserve participant anonymity.

Garden	Sampling Sites	Garden Type	Planting Method	Specimens
A	1	Educational	direct	Swan Gourd
B	1	Community	direct	Collards
C	2	Community	raised bed	Sage, String Beans
D	2	Community	raised bed	Kale (2)
E	1	Community	raised bed	Collards
G	2	Community	raised bed	Celery, Pea with Pods
H	1	Community	raised bed	Arugula
J	4	Community	raised bed	Arugula, Broccoli, Chenopod, Kale
K	2	Community	raised bed	Basil, Kale
L	2	Community	raised bed	Kale, Radish
M	2	Community	raised bed	Collards, Kale
N	2	Community	raised bed	Kale, Mint
O	2	Community	direct	Basil, Bell Pepper
P	2	Community	raised bed	Cherry Tomato, Collards
Q	1	Community	direct	Spearmint
R	3	Community	direct	Chard (2), Collards
S	2	Community	raised bed	String Beans, Tomato
T	1	Community	direct	Red Cabbage
U	1	Backyard	direct	Hot Pepper
V	1	Community	direct	Wild Chicory
W	1	Community	direct	Collards
X	2	Community	direct	Chenopod, Wild Chicory

Table 2. Soil sample results (n=38, except for % clay, where n=37).

Soil Property	Average	Min	Max	Standard Deviation	Standard Error
pH	6.93	5.63	7.54	0.42	0.07
SOM (%)	9.04	1.56	26.14	5.28	0.86
Clay (%)	9.49	3.00	22.00	4.39	0.72

Table 3. Soil trace element concentrations (mg/Kg) in sampled urban vegetable gardens (n=38).

Element	Average	Min	Max	Standard Deviation	Standard Error	Maximum Allowable
Al	7665.400	4182.000	12359.000	2285.120	370.695	n.a.
As	3.980	0.000	10.000	2.166	0.351	13[a]
Cd	0.817	0.003	7.296	1.418	0.230	2.5[a]
Co	5.480	2.000	9.000	1.595	0.259	23[b]
Cr	18.280	6.000	41.000	7.192	1.167	1; 30[c]
Cu	45.929	17.690	141.060	21.632	3.509	50[a]
Hg	0.000	0.000	0.000	0.000	0.000	0.18
Mg	5501.912	1697.490	16867.700	3385.958	549.275	n.a.
Ni	17.071	6.395	64.547	9.155	1.485	30[a]
Pb	166.344	13.793	831.315	159.272	25.837	63[a]
Ti	87.527	3.969	227.271	72.715	11.796	n.a.
V	12.370	5.000	37.000	5.169	0.839	390[b]
Zn	271.327	57.420	2738.620	442.067	71.713	109[a]

[a] NYS DEC 2006; b US EPA 2017; [c] 1 for hexavalent and 30 for trivalent species.

Table 4. Plant trace element concentrations in sampled urban vegetable gardens (n=38).

Element	Average	Min	Max	Standard Deviation	Standard Error	Maximum Allowable
Al	155.430	22.443	1560.020	305.129	49.498	1.0[a]
Cd	0.117	0.000	1.488	0.249	0.040	0.2[b]
Co	0.098	0.000	1.002	0.217	0.035	0.01[a]
Cr	0.564	0.000	3.618	0.852	0.138	2.3[b]
Cu	30.477	3.146	324.651	56.296	9.132	40[b]
Hg	0.353	0.000	3.520	0.790	0.128	0.1
Mg	4561.675	1372.220	10251.600	2248.830	364.808	n.a.
Ni	0.773	0.000	6.741	1.470	0.238	n.a.
Pb	8.984	0.000	74.717	13.929	2.260	0.3[b]
Sr	73.769	1.077	424.718	75.906	12.314	n.a.
Ti	5.188	0.000	79.904	14.651	2.377	n.a.
V	0.298	0.000	5.539	0.976	0.158	0.01[a]
Zn	76.564	20.616	308.175	53.853	8.736	0.3[a]

[a] US CDC 2019; [b] FAO/WHO Codex Alimentarius Commission, 2015.

Table 5. Highest ten values (mg/Kg) per trace element relative to plant specimen and site.

Al	**Vegetable**	**Site**	**Cd**	**Vegetable**	**Site**	**Co**	**Vegetable**	**Site**
312.00	Radish	L	0.37	Arugula	H	0.25	Arugula	H
287.50	Arugula	H	0.08	Arugula	J	0.18	Radish	L
119.67	Basil	O	0.07	Wild Chicory	X	0.09	Basil	O
84.89	Sage	C	0.07	Celery	G	0.06	Kale	M
75.90	Kale	M	0.06	Chard	R	0.04	Wild Chicory	X
39.83	Wild Chicory	X	0.05	Radish	L	0.03	Basil	K
35.03	Basil	K	0.05	Basil	O	0.03	Chenopod	X
26.82	Celery	G	0.05	Collards	P	0.02	Collards	R
26.38	Kale	K	0.05	Collards	W	0.02	Chard	Farm
24.47	Chard	R	0.03	Kale	K	0.02	String Beans	C
Cr	**Vegetable**	**Site**	**Cu**	**Vegetable**	**Site**	**Hg**	**Vegetable**	**Site**
0.90	Arugula	H	81.16	Arugula	J	0.88	Broccoli	J
0.72	Radish	L	27.57	Broccoli	J	0.41	Kale	L
0.38	Kale	M	19.89	Sage	C	0.37	Arugula	H
0.33	Basil	K	18.75	Peas with Pods	G	0.34	Radish	L
0.29	Bell Pepper	O	16.07	Arugula	H	0.34	Celery	G
0.27	Basil	O	15.86	Kale	J	0.31	Peas with Pods	G
0.26	Hot Pepper	U	13.90	Kale	L	0.25	Kale	J
0.15	Collards	M	9.33	Radish	L	0.16	Arugula	J
0.15	Cherry Tomato	P	7.00	Celery	G	0.08	Sage	C
0.11	Wild Chicory	X	4.69	Chard	R	0.00	Basil	O

Table 5. contd.

Ni	Vegetable	Site	Pb	Vegetable	Site	Sr	Vegetable	Site
1.69	Kale	D	18.68	Arugula	H	106.18	Collards	P
1.36	Arugula	H	8.04	Radish	L	52.76	Collards	W
0.58	Collards	W	6.26	Peas with Pods	G	39.01	Kale	M
0.56	Radish	L	4.01	Celery	G	34.41	Arugula	H
0.51	Kale	M	3.90	Chard	R	32.85	Red Cabbage	T
0.51	Chard	Farm	3.75	Kale	L	32.06	Collards	R
0.48	Collards	M	3.13	Sage	C	28.84	Collards	M
0.27	Collards	P	3.11	Basil	O	26.02	Arugula	J
0.24	Mint	N	2.94	Red Cabbage	T	23.85	Kale	K
0.22	Basil	O	2.78	Wild Chicory	X	23.21	Collards	B
Ti	**Vegetable**	**Site**	**V**	**Vegetable**	**Site**	**Zn**	**Vegetable**	**Site**
19.98	Arugula	H	1.38	Arugula	H	77.04	Arugula	H
9.21	Radish	L	0.50	Radish	L	45.06	Chenopod	J
3.45	Sage	C	0.23	Kale	M	41.13	Arugula	J
2.75	Kale	M	0.21	Basil	O	35.84	Collards	P
1.39	Basil	K	0.06	Kale	K	33.21	Celery	G
1.35	Kale	K	0.05	Collards	M	27.65	Basil	O
1.32	Basil	O	0.05	Kale	D	26.06	Kale	N
0.83	Collards	M	0.04	Basil	K	25.78	Mint	N
0.64	Chard	R	0.04	Mint	N	24.28	Basil	K
0.64	Mint	N	0.03	Chard	R	21.58	Broccoli	J

Table 6. Correlations between Al and trace element contents in vegetables.

Element	Pearson's r	p-value
Cd	0.436^{**}	0.006
Co	0.227	0.170
Cr	0.133	0.427
Cu	0.478^{**}	0.002
Hg	0.411^{*}	0.010
Mg	0.280	0.088
Ni	0.187	0.262
Pb	0.519^{**}	0.001
Sr	0.307	0.061
Ti	0.523^{**}	0.001
V	0.538^{**}	0.000
Zn	0.506^{**}	0.001

* Significant correlation at $p < 0.05$ (2-tailed)
** Significant correlation at $p < 0.01$ (2-tailed)

Table 7. Number of cases by trace element where vegetable trace element content exceeded that of soil.

Element	Number of Cases of Vegetable TE Content > Soil TE Content	% of Total Cases	Soil TE Content > Vegetable TE Content (% of Cases)
Al	0	0	100
Cd	2	5.3	94.7
Co	0	0	100
Cr	0	0	100
Cu	8	21.1	78.9
Hg	8	21.1	0
Mg	21	55.3	44.7
Ni	1	2.6	97.3
Pb	0	0	100
Ti	0	0	100
V	0	0	100
Zn	1	2.6	97.3
Total	39	8.6	91.4

Table 8a. Soil properties according to planting method (n = 14 for direct planting; n = 24 for planting on raised bed).

Soil Property	Statistic	Direct	Raised Bed
pH	Average	6.86	6.98
	Min	6.40	5.63
	Max	7.49	7.54
	Std Dev	0.32	0.55
	SE	0.07	0.15
SOM (%)	Average	10.39	7.09
	Min	4.80	1.77
	Max	26.14	16.85
	Std Dev	5.42	4.00
	SE	1.11	1.07
Clay (%)	Average	8.59	11.75
	Min	3.30	5.31
	Max	19.26	22.30
	Std Dev	4.17	4.83
	SE	0.85	1.29

Table 8b. Trace element levels in soil and vegetables according to planting method.

		Direct	Raised Bed	Raised Bed	Direct
	Statistic	**Soil mg/Kg**		**Vegetable mg/Kg (fresh weight)**	
Al	Average	6718.717	9494.617	80.118	194.843
	Min	4181.640	5239.670	22.443	23.165
	Max	9959.920	14662.600	478.675	1560.020
	Std Dev	1433.599	2904.417	117.241	370.685
	SE	292.632	776.238	31.334	75.666
Cd	Average	0.597	1.195	0.089	0.122
	Min	0.045	0.003	0.000	0.000
	Max	3.775	7.296	0.258	1.488
	Std Dev	0.708	2.143	0.085	0.307
	SE	0.145	0.573	0.023	0.063
Co	Average	4.656	6.962	0.069	0.110
	Min	2.397	2.813	0.011	0.000
	Max	5.973	9.715	0.358	1.002
	Std Dev	0.789	1.750	0.089	0.266
	SE	0.161	0.468	0.024	0.054
Cr	Average	19.105	17.264	0.474	0.605
	Min	6.423	6.853	0.106	0.000
	Max	41.233	28.711	1.592	3.618
	Std Dev	7.721	6.346	0.507	1.011
	SE	1.576	1.696	0.135	0.206
Cu	Average	44.826	41.336	11.232	41.564
	Min	21.794	17.690	3.441	3.146
	Max	75.584	66.987	18.750	324.651
	Std Dev	15.015	14.604	5.674	68.784
	SE	3.065	3.903	1.517	14.040
Mg	Average	4845.997	6720.741	5181.821	4462.966
	Min	2188.790	1697.490	1518.560	1372.220
	Max	10062.800	16867.700	10251.600	9889.980
	Std Dev	2361.422	4497.373	2624.998	2094.983
	SE	482.023	1201.974	701.560	427.637
Ni	Average	14.732	21.252	0.515	0.979
	Min	6.645	6.395	0.000	0.000
	Max	30.756	64.547	2.320	6.741
	Std Dev	4.891	12.987	0.765	1.770
	SE	0.998	3.471	0.204	0.361

Table 8b. contd.

Pb	Average	136.907	213.790	5.445	10.792
	Min	13.793	21.463	0.000	0.000
	Max	398.272	831.315	15.607	74.717
	Std Dev	92.766	231.851	4.835	16.982
	SE	18.936	61.965	1.292	3.466
Sr	Average	50.988	36.128	61.680	80.087
	Min	19.565	13.077	2.904	1.077
	Max	143.517	70.902	211.039	424.718
	Std Dev	32.285	19.524	60.615	84.432
	SE	6.590	5.218	16.200	17.235
Ti	Average	123.906	24.772	1.360	7.396
	Min	8.875	-1.496	0.000	0.000
	Max	227.271	93.763	5.269	79.904
	Std Dev	66.678	23.902	1.532	18.172
	SE	13.611	6.388	0.409	3.709
V	Average	13.393	10.595	0.085	0.420
	Min	5.042	8.043	0.000	0.000
	Max	36.996	13.750	0.825	5.539
	Std Dev	6.213	1.580	0.219	1.210
	SE	1.268	0.422	0.058	0.247
Zn	Average	232.480	338.966	51.208	89.995
	Min	59.390	57.420	29.804	20.616
	Max	1006.120	2738.620	110.589	308.175
	Std Dev	187.817	696.899	20.871	62.658
	SE	38.338	186.254	5.578	12.790